Effects of elevation and land use on the biomass of trees, shrubs and herbs at Mount Kilimanjaro

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Abstract. The protection and sustainable management of forest carbon stocks, particularly in the tropics, is a key factor in the mitigation of global change effects. However, our knowledge of how land use and elevation affect carbon stocks in tropical ecosystems is very limited. We compared aboveground biomass of trees, shrubs and herbs for eleven natural and human-influenced habitat types occurring over a wide elevation gradient (866–4550 m) at the world’s highest solitary mountain, Mount Kilimanjaro. Thanks to the enormous elevation gradient, we covered important natural habitat types, e.g., savanna woodlands, montane rainforest and afro-alpine vegetation, as well as important land-use types such as maize fields, grasslands, traditional home gardens, coffee plantations and selectively logged forest. To assess tree and shrub biomass with pantropical allometric equations, we measured tree height, diameter at breast height and wood density and to assess herbaceous biomass, we sampled destructively. Among natural habitats, tree biomass was highest at intermediate elevation in the montane zone (340 Mg ha\(^{-1}\)), shrub biomass declined linearly from 7 Mg ha\(^{-1}\) at 900 m to zero above 4000 m, and, inverse to tree biomass, herbaceous biomass was lower at mid-elevations (1 Mg ha\(^{-1}\)) than in savannas (900 m, 3 Mg ha\(^{-1}\)) or alpine vegetation (above 4000 m, 6 Mg ha\(^{-1}\)). While the various land-use types dramatically decreased woody biomass at all elevations, though to various degrees, herbaceous biomass was typically increased. Our study highlights tropical montane forest biomass as important aboveground carbon stock and quantifies the extent of the strong aboveground biomass reductions by the major land-use types, common to East Africa. Further, it shows that elevation and land use differently affect different vegetation strata, and thus the matrix for other organisms.

Key words: carbon stock; elevation gradient; land-use change; tropical mountain; vegetation strata.

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INTRODUCTION

Tropical forests are estimated to harbour some 25% of the terrestrial biosphere’s carbon (Bonan 2008). They play a key role in ecosystem carbon sequestration and hence, for climate change mitigation (Malhi and Grace 2000, Lewis et al. 2004, Houghton 2007). While there is substantial
evidence that primary tropical forests are a carbon sink, large uncertainties exist regarding the size of standing stocks and the impacts of human land use on them, especially in Africa (Houghton 2007, Lewis et al. 2009, Pan et al. 2011). These uncertainties prevent effective estimates of anthropogenic carbon losses when deforestation and land use occur. Such estimates, however, are essential to conserve and sustainably manage tropical forest carbon stocks, e.g., in programs for the reduction of emissions due to deforestation and degradation (REDD+; Gibbs et al. 2007).

Local forest inventories are lacking for various regions and habitat types, including tropical montane ecosystems, for which the relationship between elevation and carbon stocks is poorly characterised (Houghton 2005, Spracklen and Righelato 2013). Previous studies of tropical montane biomass stocks have been conducted in tropical Asia (Kitayama and Aiba 2002, Culmsee et al. 2010), South America (Alves et al. 2010, Girardin et al. 2010, Moser et al. 2011) and East Africa (Marshall et al. 2012). In most of these studies, carbon stocks decreased with elevation, while carbon stocks at a given elevation varied strongly between and within continents. This pattern is likely to be driven by differences in climate. However, other environmental parameters, such as soil nutrient availability, also play a role, and general patterns still need to be identified (Malhi et al. 2006, Slik et al. 2010, Spracklen and Righelato 2013). Also, most studies did not consider shrub or herb biomass, whose importance relative to trees is therefore poorly known.

Land-use change, particularly deforestation, contributes importantly to the land-atmosphere carbon flux (Foley et al. 2005, Houghton 2007, Ramankutty et al. 2007). In tropical Africa, the carbon emissions from land-use change exceed those from fossil fuel combustion (Canadell et al. 2009) and future land-use change might further deplete carbon stocks, as some land-use types still store significant amounts of carbon (Albrecht and Kandji 2003, Kumar and Nair 2011). However, accurate estimates of land-use change effects on carbon stocks are not available for many tropical habitats and the effect of land use on woody and non-woody vegetation strata has not been quantified. This makes it difficult to predict how ecosystem carbon budgets will respond to future changes (Gibbs et al. 2007, Houghton 2007).

Here we study the aboveground biomass (hereafter biomass) of natural and anthropogenically affected tropical habitat types at different elevations at Mt. Kilimanjaro, Tanzania. Mt. Kilimanjaro is the world’s highest free-standing mountain and harbours a wide range of different habitat types, including savanna woodlands, mountain cloud forest and afro-alpine vegetation. Its vegetation has been strongly influenced by human impacts and there is considerable variation in land-use types and intensities across the mountain (Agrawala et al. 2003, Hemp 2006a). We measured the biomass of three vegetation strata (tree, shrub and herbaceous layer) in six important major natural habitat types and five anthropogenically affected types over an elevation range of 3680 m from the savanna to the afro-alpine zone based on forest inventories and destructive herbaceous biomass sampling. Because tree architecture varies strongly between habitat types as different as tropical rainforest and savannas, we used different allometric equations to adequately estimate tree and shrub biomass from our forest inventory data. We asked how biomass of natural habitats is distributed along the elevation gradient and how anthropogenic influence modulates the biomass at different elevations.

**METHODS**

**Study system and design**

We studied the southern and south-eastern slopes of Mt. Kilimanjaro, where climate gradients are most pronounced and various habitat types are present. Mean annual temperature ranges from about 23°C at 800 m a.s.l. in the savanna to −7°C at 5895 m at Uhuru-peak (Hemp 2006b). Precipitation also strongly changes with elevation with dry foothills and afro-alpine heathlands, and a mid-elevational precipitation peak at around 2200 m (Hemp 2006b). An optimum for biomass production may thus be expected at medium elevations, where precipitation is high, but temperatures are not too low to constrain growth considerably.

The following important natural and anthropogenically used habitat types occur at the
mountain and were investigated in our study: In
the foothills of the mountain from 800 to 1100 m,
savanna woodlands with *Acacia-Commiphora*
vegetation dominate the natural landscape.
However, these woodlands are increasingly
transformed into maize fields for local and
regional food production.

In the densely populated lower-montane area
between 1200 and 2000 m, patches of lower
montane forest represent the natural vegetation
and three major land-use types are found in this
elevation zone. The traditional agricultural sys-
tem of the local Chagga people, the “Chagga
home gardens”, is by far the most abundant. The
Chagga people use these multi-layered gardens to
grow different crops such as bananas, coffee,
avocados, taro and beans, keeping forest trees for
shade (Hemp 2006a). The second land-use type is
grasslands, which are cut frequently by the
Chagga people to obtain fodder for their live-
stock. More recently, commercial coffee planta-
tions have spread on the southern slope of
Kilimanjaro. These plantations mostly cover
quite large and homogeneus areas with scat-
tered, often non-native, trees.

The natural montane *Ocotea* forest grows
between 2100 and 2800 m and is dominated by
the camphor tree (*Ocotea usambarensis*, Laurae-
ceae), which was also the main target of
commercial selective logging activities until
1984 (Agrawala et al. 2003). The legacy of this
forestry is seen in *Ocotea* forests disturbed by
selective logging.

Above 2800 m, the gymnosperm *Podocarpus
latifolius* (Podocarpaceae) dominates the upper
montane *Podocarpus* forest up to about 3100 m. In
the subalpine zone, up to 4000 m, patches of natural
*Erica* forest dominated by *E. trimera* represent remnants of Africa’s highest forests.

Finally, in the alpine zone up to 4500 m, cushion
plants of the genus *Helichrysum* dominate to-
gether with tussock grasses.

Across the slope, plots were established in the
11 habitat types described above so that the most
important primary and human dominated hab-
itats were represented (Hemp 2006b). Each
habitat type was replicated five times, resulting
in 55 plots for this study. Each plot was 0.25 ha in
size, except for savanna plots, which were one
hectare in size on account of their low tree
density. Plot boundary positions were recorded
with GPS and permanently marked with subterr-
anean iron nails. For each plot, mean annual
temperature (MAT) was obtained from in-situ
measures by temperature loggers (data logger
DK320, Driesen and Kern GmbH, Bad Bramstedt,
Germany) and mean annual precipitation (MAP)
was modeled using long-term observations
based on a 15-year dataset from a network rain
gauges distributed across the whole mountain
(Appelhans et al. 2014; see also Appendix A for
further details on habitat types and plot charac-
teristics).

**Measurements**

**Tree inventory.**—Within each plot, all trees
wider than 10 cm diameter at breast height
dbh) were marked with aluminium tags and
their dbh and height were measured. The dbh
was measured with a diameter tape (Forestry
Suppliers, USA) at 1.3 m for normally shaped
trees and 20 cm below or above when branches
or irregular shapes impeded measurement at that
height. The 1.3 m height was measured from the
highest ground level around the stem to stan-
dardize measurements taken on slopes. For trees
which were strongly buttressed or too big to
measure by hand, a laser dendrometer (Criterion
RD 1000 with TruPulse 200/200, Centennial,
USA) was used to measure the tree above the
buttresses and at 1.3 m. Lianas above 10 cm in
diameter were also marked and their dbh was
measured. Tree height was measured using an
ultra-sonic hypsometer (Vertex IV Hypsometer,
Haglöf, Langsele, Sweden) or a laser rangefinder
(TruPulse 200/200). The tree inventories were
carried out between December 2010 and March
2013 for all plots containing trees except for the
five plots in the subalpine *Erica* forest, where we
only studied herbaceous biomass.

**Shrub inventory.**—The shrub inventory was
carried out within a 5 × 20 m subplot in the
centre of each plot. Within this subplot, the shrub
layer was defined as consisting of all woody
stems exceeding 1.3 m in height, but below 10 cm
dbh and thus not included in the tree inventory.
We measured dbh at 1.3 m with a diameter tape
(Forestry Suppliers; for dbh’s above 3 cm) or a
caliper (for dbh’s below 3 cm) and the height of
each shrub with a hypsometer.

**Herbaceous biomass collection.**—For the herba-
ceous biomass, four 0.25-m² samples per plot
were taken non-randomly from areas were the herbaceous layer was considered as being representative of the whole plot. The biomass of the herbaceous layer (hereafter herbaceous biomass) included forbs with a woody stem, mosses and lichens, which were collected from ground level using a wooden frame of 50×50 cm and scissors. Samples were dried in a drying oven at 72±8°C for 72 hours and then weighed. In habitat types with pronounced rainfall seasons (savanna, maize fields, grasslands, coffee plantations, home gardens), biomass was collected after the wet season maximum, in December and January 2010–2011, while in the maize plots, it was collected in June and July 2012, shortly before the harvest. The other samples were collected between December 2010 and October 2012. This allowed us to estimate the maximum standing biomass of the herbaceous layer on each plot.

Wood density.—Wood density was measured as dry weight divided by fresh volume for field-collected wood cores (Increment borer 50 cm, Suunto, Vantaa, Finland) of the dominant tree and shrub species accounting for 80% of the ground cover of all trees and shrubs in each plot, i.e., for 66 of a total of 132 identified species. Wood cores were taken from at least five individuals, if possible from different sites, at 1.3 m height and samples were oven dried at 72°C for at least 72 hours and weighed under exclusion of any remoistening (in sealed boxes with silica gel). Wood density values for the remaining 66 species were taken from the global wood density database (Zanne et al. 2009). When species-level data was not available or species identification was not possible (110 out of 4500 stems), mean genus-level values were used and family-level means were applied when the genus-level data was not available or when an individual could not be identified to genus level (74 out of 4500 stems). If an individual could not be identified at all (250 out of 4500 stems), mean plot values were used (Marshall et al. 2012).

Allometric equations
The choice of an appropriate allometric model is the crucial step towards minimizing the errors in forest biomass estimates (Chave et al. 2004, Molto et al. 2013). Therefore, we carefully chose the allometric equations most suited for the 4500 stems measured in the different habitat types so as to acquire the best possible approximation of their aboveground biomass (Table 1; Appendix B). For the forest habitats and lower montane land-use plots, we used pantropical equations for trees, shrubs and lianas developed by Chave et al. (2005) and by Schnitzer et al. (2006), which were specifically developed for use outside their sample range and were shown to produce

### Table 1. Allometric equations used for the different habitat types and plant groups.

<table>
<thead>
<tr>
<th>Type</th>
<th>Allometric equation</th>
<th>Reference</th>
<th>Plots</th>
</tr>
</thead>
<tbody>
<tr>
<td>Savanna</td>
<td>AGB = 0.0763 × dbh^{2.2016} × H^{0.4918}</td>
<td>Mugasha et al. 2013</td>
<td>Savanna Maize fields Low. montane forest Home gardens (1, 3) Coffee plantations (2) Ocotea forest Select. logged forest Polycarpus forest</td>
</tr>
<tr>
<td>Wet forest</td>
<td>AGB = exp(−2.557 + 0.940 (ln(wd × dbh^2 × H)))</td>
<td>Chave et al. 2005</td>
<td>Home gardens (2, 4–5) Coffee plant. (1, 3–5) Grasslands</td>
</tr>
<tr>
<td>Dry forest</td>
<td>AGB = exp(−2.187 + 0.916 (ln(wd × dbh^2 × H)))</td>
<td>Chave et al. 2005</td>
<td>Home gardens Ocotea forest Select. logged forest</td>
</tr>
<tr>
<td>Lianas</td>
<td>AGB = exp(−1.484 + 2.657 (ln(dbh)))</td>
<td>Schnitzer et al. 2006</td>
<td>Low. montane forest</td>
</tr>
<tr>
<td>Bananas</td>
<td>AGB = 0.03 × dbh^{2.13}</td>
<td>Hairiah et al. 2010</td>
<td>Home gardens Coffee plantations</td>
</tr>
<tr>
<td>Coffee</td>
<td>AGB = 0.281 × dbh^{2.06}</td>
<td>Hairiah et al. 2010</td>
<td>Home gardens Coffee plantations</td>
</tr>
</tbody>
</table>

Notes: Type denotes the habitat type for which equations apply for, according to their authors. Plots denotes the types of habitat (and for home gardens and coffee plantations also the exact plots in parentheses) for which the equations were used in our study. Measured parameters used in the equations were total plant height (H), diameter at breast height (dbh) and wood density (wd, measured as dry weight divided by fresh volume). For a detailed description of the choice of the equations, see Appendix B.
reliable estimates for tropical forests and agricultural systems in Africa (Kuyah et al. 2012, Vieilledent et al. 2012, Fayolle et al. 2013). For bananas and pruned coffee shrubs, we opted for the equations developed by Hairiah et al. (2010). For the savanna, and for the maize fields with their occasional trees, we selected an equation recently developed by Mugasha et al. (2013), because it was specifically developed for dry woodlands in Tanzania (Table 1). A detailed description of the choice of the equations for each habitat type is given in the Appendix B. We added the biomasses per plot and upscaled them to values for one hectare. The carbon content of woody and herbaceous biomass was approximated as 48.2% (Table 2; Thomas and Martin 2012).

**RESULTS**

**Biomass of natural habitats over elevation**

Total aboveground biomass of natural habitats followed a clear unimodal pattern across the elevation gradient, peaking at around 2200 m in the montane Ocotea forest ($R^2 = 0.86, F_{1,19} = 6.6, P = 0.019$; Fig. 1). The highest biomass was found in one of the lower montane forest plots, which reached 664.3 Mg ha$^{-1}$ (Appendix A). Total biomass decreased at both ends of the elevation gradient, reaching 10.4 Mg ha$^{-1}$ in the savannas and 6.3 Mg ha$^{-1}$ in the treeless, alpine Helichrysum vegetation. Precipitation explained a very similar amount of the variation in total biomass (Fig. 2; quadratic regression, $R^2 = 0.85, F_{1,20} = 28.2, P < 0.001$) as elevation did (Fig. 1; see above).

Trees contributed 33% to total biomass in the savanna, 98% in the forest and 0% in the alpine zone. Similar to total biomass, tree biomass showed a hump-shaped pattern across the elevation gradient ($R^2 = 0.89, F_{1,19} = 10.6, P = 0.004$; Fig. 1). Individual tree biomass values ranged from 0.002 Mg to a 24.4 Mg Entandrophragma excelsum tree in the lower montane forest. However, 88% of the trees weighed less than one ton and mean tree biomass was 0.6 Mg. The smallest tree biomass per plot was found in the savanna woodlands, with a mean of 3.5 Mg ha$^{-1}$.

Shrub biomass decreased significantly with elevation ($R^2 = 0.47, F_{1,21} = 20.6, P < 0.001$; Fig. 1). Fitting a quadratic function of shrub biomass

<table>
<thead>
<tr>
<th>Zone</th>
<th>Type</th>
<th>Biomass (Mg ha$^{-1}$)</th>
<th>Total carbon (Mg C ha$^{-1}$)</th>
<th>Zone</th>
<th>Type</th>
<th>Biomass (Mg ha$^{-1}$)</th>
<th>Total carbon (Mg C ha$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Savanna</td>
<td>Savanna</td>
<td>2.5 (0.6)</td>
<td>5.1 (1.0)</td>
<td>Lower montane</td>
<td>Natural forest</td>
<td>0.9 (0.3)</td>
<td>1.74 (0.42)</td>
</tr>
<tr>
<td></td>
<td>Maize field</td>
<td>15.2 (2.9)</td>
<td>174.0 (42.8)</td>
<td></td>
<td>Home garden</td>
<td>0.7 (0.2)</td>
<td>44.9 (8.3)</td>
</tr>
<tr>
<td></td>
<td>Coffee plantation</td>
<td>1.6 (0.4)</td>
<td>49.9 (8.3)</td>
<td></td>
<td>Coffee plantation</td>
<td>1.6 (0.4)</td>
<td>49.9 (8.3)</td>
</tr>
<tr>
<td></td>
<td>Grassland</td>
<td>2.9 (1.1)</td>
<td>17.0 (1.7)</td>
<td>Montane</td>
<td>Ocotea forest</td>
<td>1.1 (0.1)</td>
<td>135.7 (23.6)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Logged forest</td>
<td>1.3 (0.2)</td>
<td>172.5 (10.6)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td>Upper montane</td>
<td>Podocarpus forest</td>
<td>4.0 (1.5)</td>
<td>179.5 (21.6)</td>
</tr>
<tr>
<td>Subalpine</td>
<td>Erica forest</td>
<td>8.8 (2.1)</td>
<td>33.2 (2.1)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Alpine</td>
<td>Helichrysum zone</td>
<td>6.3 (2.1)</td>
<td>3.1 (1.0)</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes: Biomass is given for the herbaceous (Herbs), shrub (Shrubs) and tree layer (Trees) and as total. For definition of the different habitat types, elevation zones and sampling design, see Materials and methods. Natural habitats are in boldface. Mg C denotes total carbon (48.2% of total biomass; Thomas and Martin 2012).
with elevation explained a similar proportion of the total variation \( R^2 = 0.52 \) for the quadratic versus 0.47 for the linear function), but did not improve the AIC. Shrub biomass exceeded tree biomass in the savanna (42\% of total biomass), but made up only 1.3\% of the total biomass in the lower montane rainforest, 2.2\% in the montane Ocotea forest and 0.5\% in the upper Podocarpus forests. The highest shrub biomass was found in a montane Ocotea forest plot with 10 Mg ha\(^{-1}\) (Appendix A).

In natural habitats herbaceous biomass showed bimodal pattern along the elevation gradient \( R^2 = 0.48, F_{1,26} = 10.7, P = 0.003; \) Fig. 1), which was inverse to the hump shaped elevation pattern of tree biomass (reflecting a negative relationship between tree and herbaceous biomass: \( F_{1,21} = 6.4, P = 0.02 \)). Herbaceous biomass contributed 24\% of the total biomass in the savanna, 0.2\% in the lower montane forest.

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**Fig. 1.** Relationships of total (solid line), tree (dashed line), shrub (dash-dotted line) and herbaceous biomass (dotted line) of natural habitat types over the elevation gradient at Mount Kilimanjaro. Total biomass: cubic function \( R^2 = 0.86, F_{1,19} = 6.6, P = 0.019 \), Trees: cubic function: \( R^2 = 0.89, F_{1,19} = 10.6, P = 0.004 \); Shrubs: linear function: \( R^2 = 0.47, F_{1,21} = 20.6, P < 0.001 \); Herbaceous: cubic function: \( R^2 = 0.48, F_{1,26} = 10.7, P = 0.003 \). Please note the log-scale.

**Fig. 2.** Relationship between annual precipitation and total aboveground biomass per plot on a log-log scale for natural habitat types over the elevation gradient at Mount Kilimanjaro. Dashed lines indicate 95\% CI. ***: \( P < 0.001 \).
0.4% in the *Ocotea* forest, 1% in the upper montane *Podocarpus* forest and 100% in the treeless alpine vegetation (Fig. 1). The lower montane forests had the lowest herbaceous biomass of 0.9 Mg ha\(^{-1}\). The highest herbaceous biomass was found in the subalpine *Erica* forest with up to 16.7 Mg ha\(^{-1}\). The alpine *Helichrysum* vegetation above 4000 m still harboured substantial amounts of herbaceous biomass (mean 6.3 Mg ha\(^{-1}\)).

### Anthropogenic effects on biomass at different elevations

In the savanna zone, while tree and shrub biomass were significantly lower in maize fields than in savanna woodlands (trees: \(F_{1,7} = 8.75, P = 0.02\); shrubs: \(F_{1,7} = 11.87, P = 0.01\)), herbaceous biomass was more than 5-fold higher in maize fields than in savannas (\(F_{1,7} = 32.91, P < 0.001\); Fig. 3a). As a result, total biomass did not differ between savanna woodlands and maize fields.

In the lower montane zone, total biomass strongly differed across the four different land-use types (\(F_{3,12} = 8.44, P < 0.001\)), declining strongly from natural forests to home gardens and coffee plantations and reaching its lowest value in grasslands, where it was 99% lower than in natural forests (Fig. 3b). While home gardens and coffee plantation stored more biomass than grasslands, they did not differ significantly from each other (\(F_{1,7} = 1.4, P = 0.27\)). Tree biomass followed the same pattern as total biomass (Fig. 3b). Shrub biomass also differed significantly across habitat types (\(F_{3,12} = 41.03, P < 0.001\)), but was significantly lower than in natural lower montane forests only in grasslands (\(F_{1,7} = 43.07, P < 0.001\); Fig. 3b). Also, shrub biomass was higher in home gardens than in natural forest and coffee plantations, mainly due to the abundant bananas in the home gardens. The herbaceous biomass...
did not differ between natural lower montane forests, home gardens and coffee plantations, but was higher in grasslands than in forests and home gardens (Fig. 3b).

In the montane Ocotea zone, selectively logged montane Ocotea forests did not differ from undisturbed ones in their tree, shrub, herbaceous and total biomass (Fig. 3c).

**DISCUSSION**

**Biomass of natural habitats over elevation**

The distinct unimodal pattern in total and tree biomass over elevation coincided strongly with the precipitation pattern (Fig. 2), suggesting that precipitation is the main determinant of biomass variation at Mt. Kilimanjaro. This is in line with a meta-analysis across larger spatial scales (Stegen et al. 2011). Nevertheless, further drivers may modulate the precipitation effect on biomass, including soil nutrient relations or biotic interactions, such as herbivory (Stegen et al. 2011, Metcalfe et al. 2014).

Other studies on tropical mountains found a monotonous decrease of biomass with elevation rather than a unimodal pattern, e.g., in tropical Asia (Kitayama and Aiba 2002) and the Andes (Girardin et al. 2010, Moser et al. 2011), or even an increase (e.g., Culmsee et al. 2010, Marshall et al. 2012). However, in the cases where biomass decreased monotonously with elevation, the lowlands were not dryer than the higher elevations, contrasting with the precipitation pattern at Mt. Kilimanjaro. Moreover, the studies reporting an increase in biomass with elevation did not extend higher than 2500 m, and thus presumably did not exceed the tropical tree line.

In a remote sensing study, Willcock et al. (2012) approximated woody savanna biomass close to Mt. Kilimanjaro to 1–10 Mg ha⁻¹, which is congruent with our estimate of 7.9 Mg ha⁻¹ for the combined mass of trees and shrubs. Other biomass studies of slightly more humid savanna ecosystems arrived at slightly higher estimates, e.g., 28–60 Mg ha⁻¹ for the Miombo woodlands in southern Tanzania (Shirima et al. 2011) and 6.5–82.1 Mg ha⁻¹ for savannas in South Africa (Colgan et al. 2012). The Miombo woodlands with twice the rainfall of the savannas surrounding Mt. Kilimanjaro are more forest-like with trees higher than 20 m, whereas Mt. Kilimanjaro’s savanna trees are sparser and rarely exceed 8 m (10 out of 234 stems in our study). An additional reason for the comparatively low biomass of Mt. Kilimanjaro’s savannas may be the high logging and burning pressure even for the savanna remnants studied here, which has intensified with the strong population increase in the region (Agrawala et al. 2003).

The lower montane forest zone at Mt. Kilimanjaro had the highest plot-level total biomass across the whole mountain, and also the highest variability between plots (SD of 198.5 Mg ha⁻¹). This was due to the patchy distribution of trees of >70 cm dbh, which comprised over 70% of tree biomass where present, as in a pantropical review (Slik et al. 2013). This patchy distribution of large trees may have been further increased by illegal timber and firewood cutting, which occurs more frequently in the lower montane forests due to their easier accessibility and proximity to the national park border just below.

Tree biomass remained high all the way up to the upper montane Podocarpus forest at 2800 m with a mean of 372 Mg ha⁻¹ (Table 2). Other studies in high-altitude tropical regions report 129 and 112 Mg ha⁻¹ at ~3000 m in the Andes (Girardin et al. 2010, Moser et al. 2011) and 210 Mg ha⁻¹ in tropical Asia (Kitayama and Aiba 2002). This suggests that the general pattern found for lowland rainforests, according to which forest biomass is higher in Africa than in tropical America (Banin et al. 2012, Lewis et al. 2013, Slik et al. 2013), may also hold true for montane forests. The biomass of the upper montane Podocarpus forests on Kilimanjaro actually comes close to the estimates for African-wide lowland rainforest biomass (395.7 Mg ha⁻¹) compiled by Lewis et al. (2013). However, we are not aware of any other study investigating the biomass of African high montane forests; thus future studies need to confirm the generality of high carbon stocks in African upper-montane forests (Spracklen and Righelato 2013).

Shrub biomass was highest in the montane Ocotea forest with about 6 Mg ha⁻¹ and differed strongly between habitat types. To date, few studies have separately estimated shrub biomass in tropical forests, especially along elevation gradients. E.g., Culmsee et al. (2010) found a slight increase of biomass of small diameter stems (between 2 and 10 cm dbh) with elevation,
which contrasts with our pattern (Fig. 1). In our study, herbaceous understorey growth was strong in the upper montane Podocarpus forests, where herbaceous vegetation reached heights of 2 m and more and exceeded the biomass of the shrub layer (Fig. 1). As these herbs compete with shrubs for light in the dark understorey, the strong herb layer might explain the relatively low shrub biomass in Mount Kilimanjaro’s upper montane Podocarpus forests.

Herbaceous biomass was inversely related to woody biomass (Fig. 1), a pattern that seems largely due to reduced light availability under closed canopies. Thus, herbs only played a substantial role for total biomass production in habitats with low tree density.

The highest values of herbaceous biomass were found for subalpine Erica forest, reaching 16 Mg ha\(^{-1}\). This may be due to the relatively open canopy of these forests, which allows for an extensive moss and lichen layer sometimes more than 20 cm thick. Few biomass estimates have been made for tropical alpine regions. In the South American Andes, biomass stocks of the Páramo and Puna vegetation, where trees are also scarce, were reported between 6.7 and 7.4 Mg ha\(^{-1}\) (Gibbon et al. 2010, Oliveras et al. 2014). Our estimates of 6.3 Mg ha\(^{-1}\) for the alpine Helichrysum heaths are the first for afro-alpine vegetation.

**Anthropogenic effects on biomass at different elevations**

In the savanna zone, tree and shrub biomass was much lower in maize fields than in savannas, while the herbaceous biomass (i.e., crop biomass) was significantly higher, resulting in no net difference in total biomass (Fig. 3a). However, as crop biomass is harvested every year, it is an ephemeral carbon stock, and thus the conversion of savanna to maize fields may result in substantial net carbon losses in the longer term (Woomer 1993). Furthermore, agricultural intensification may severely alter soil conditions in savannas, resulting in significant soil carbon losses (Williams et al. 2008). On the other hand, abandoned crop fields may regrow quickly and restore the lost aboveground carbon within \(~30–60\) years ago. However, although biomass may recover quickly, selectively logged forests may still continue to differ in their structure, species richness and composition (Marín-Spiotta et al. 2007, Martin et al. 2013, Rutten et al. 2015).

**Conclusions**

The coinciding unimodal elevation patterns of aboveground biomass and precipitation suggest that biomass is mainly driven by precipitation at Mt. Kilimanjaro. As tree biomass contributed 98\% of total biomass in forest habitats, shrub and herb biomass may be largely neglected for forests, if conserving carbon stocks is the only interest. However, in open habitats, such as systems and grasslands at the same elevation indicates that land-use change in the lower montane zone results in considerable carbon losses (Fig. 3b). Tree reduction seemed to be the main driver for these losses, and traditional home gardens and coffee plantations still stored 26\% and 16\% of the natural forests’ biomass, respectively, while mainly treeless grasslands retained only 1\% of the forest biomass. Not all vegetation strata were negatively affected. For example, home gardens had a higher biomass in the shrub layer than natural forests and coffee plantations. This was due to the high density of banana plants, which are short-lived and hence, also constitute an ephemeral carbon stock. Accordingly, home gardens store higher amounts of carbon than traditional grasslands, but not than shaded coffee plantations. This supports conclusions that carbon stocks in agricultural landscapes depend largely on the abundance and size of woody plants (for reviews, see Albrecht and Kandji 2003, Luedeling et al. 2011).

Past selective logging of large individual trees neither affected the current total biomass of montane Ocotea forests, nor did it significantly affect the biomass of the different strata (Fig. 3c). In accordance with our results, Medjibe et al. (2011) found that selective logging and thinning of trees does not strongly affect carbon stocks in central African lowland forests and Gourlet-Fleury et al. (2013) found a full recovery of aboveground biomass after selective logging within 24 years. At Mt. Kilimanjaro, selective logging mainly concerned the commercially important camphor tree (Ocotea usambarensis) and the logging activities in this zone occurred \(30–60\) years ago. However, although biomass may recover quickly, selectively logged forests may still continue to differ in their structure, species richness and composition (Marín-Spiotta et al. 2007, Martin et al. 2013, Rutten et al. 2015).
savannas, shrub biomass can exceed tree biomass and thus, leaving out the shrub layer in inventories will clearly fail to produce accurate estimates of aboveground carbon stocks. Compared with tree biomass, herbaceous biomass followed an inverse elevation pattern and played a considerable role in open habitats, particularly in the higher subalpine and alpine vegetation.

Biomass stocks were highest in the montane forest belt that mostly lies inside Mt. Kilimanjaro National Park. Consequently, truly protecting these forests together with the forest remnants in the lower montane area and the savannas is by far the most effective means of reducing carbon emissions due to deforestation in this zone. This is underlined by our finding that upper montane $Podocarpus$ forests on Mt. Kilimanjaro store considerably more carbon than forests at similar elevations on other continents. Our study furthermore provides first estimates of afro-alpine carbon stocks.

We showed that the transformation of savannas into intensive maize fields and of natural lower montane forests into agricultural landscapes is a major cause of aboveground carbon loss on Mt. Kilimanjaro. However, multi-layered agroforestry systems, such as the traditional home gardens, can still store up to a quarter of the carbon of the natural forests, underlining the importance of agroforestry for maintaining carbon stocks. Furthermore, we showed that land use differently affects the different vegetation strata and can substantially change the proportion of woody and non-woody components in an ecosystem. This may be important for other ecosystem properties, such as productivity, soil formation and biodiversity.

Tanzania was chosen for a pilot study for calculating REDD+ opportunities to prevent carbon losses in developing countries (Burgess et al. 2010). In this context, Mt. Kilimanjaro provides an excellent example of a mountain densely populated up to around 2000 m and with a prominent role for regional timber and water supply (Agrawala et al. 2003). Protecting the forest in the national park above about 2000 m and maintaining large shading trees in home gardens and coffee plantations at lower elevations, along with protecting the few lower montane forest remnants and the savanna remnants, may therefore be the most effective way of minimizing carbon losses due to land-use intensification at this mountain.

Acknowledgments

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Literature Cited


7:e44795.


SUPPLEMENTAL MATERIAL

APPENDIX A

Table A1. Overview over biomass and carbon stocks of the 55 study plots, their location (Zone), whether they are anthropogenically affected (Status), elevation, and climate type.

<table>
<thead>
<tr>
<th>ID</th>
<th>Zone</th>
<th>Status</th>
<th>Plot</th>
<th>Elevation (m)</th>
<th>Climate type</th>
<th>Biomass (Mg ha⁻¹)</th>
<th>Total carbon (Mg C ha⁻¹)</th>
</tr>
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<td>Dry</td>
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<td>Wet</td>
<td>1.47</td>
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</table>
Choice of the allometric equations

The pantropical allometric equations developed by Chave et al. (2005) have been widely applied for natural and disturbed tropical forests across all continents (e.g., Djomo et al. 2010, Preece et al. 2012, Rutishauser et al. 2013), and estimates were found not to be significantly biased compared with actual biomass values or locally developed equations (Vieilledent et al. 2012, Fayolle et al. 2013). Several of these studies were conducted in tropical Africa (e.g., Djomo et al. 2010, Fayolle et al. 2013), confirming the utility of the approach for Africa, although no samples from Africa had been used by Chave et al. (2005) when developing the models. We assigned our plots according to their local temperature and precipitation to the categories “dry forest”, “moist forest” and “wet forest” as used by Chave et al. (2005) (Table B1, Appendix A). These categories are based on the ratio between mean annual temperature (MAT) and mean annual precipitation (MAP) with MAT: MAP between 1:25 and 1:50 corresponding to dry forest (subhumid), between 1:50 and 1:100 to moist forest (humid) and < 1:100 to wet forest (per-humid).

Kuyah et al. (2012) showed that Chave’s “dry forest” equation estimated the biomass of East African agroforestry systems in humid conditions with a mean error of 5%, whereas the “moist forest” equation estimated it with 25% error. Therefore, we also used the dry forest equation for the three agriculturally used land cover types of the lower montane zone (home gardens, coffee plantations and grasslands), although climatically they would be assigned to the “moist forest” conditions. To test whether this use of the dry forest equation affected our results, we repeated our analyses with the moist forest equation and found that it did not change our conclusions concerning the differences between land cover types in the lower montane zone, although it yielded somewhat higher tree biomass estimates. Furthermore, it has been recently shown that crown diameter may be an additional crucial parameter improving allometric biomass estimates (Goodman et al. 2014). Although data on crown diameter was available for our trees, it could not be implemented, as there is currently no pantropical equation including crown size available with a reasonable sample size across
continents (Goodman et al. 2014). For lianas we used the pantropical equation by Schnitzer et al. (2006) (Table B1).

The biomass of banana plants and regularly pruned coffee plants in coffee plantations and home gardens was obtained using the equations recommended by Hairiah et al. (2010), all using dbh as the only input variable (Table B1).

Savanna trees in this study were generally very small in diameter and the Chave equations have been shown to be less precise for small diameter trees (van Breugel et al. 2011). Therefore, we adopted a more recent equation for the trees in the savanna zone (SAV, MAI; Appendix A), which was especially developed by Mugasha et al. (2013) for savanna and Miombo woodlands in Tanzania (Table B1). Mugasha et al. (2013) developed their equations based on destructive sampling over four savanna regions across Tanzania and also showed that the Chave “dry forest” equation underestimated the aboveground biomass of the Miombo woodlands with an error of 11%.

Overall, we are confident that we applied the most reliable equations to estimate biomass for the habitats of our study.

Table B1. Allometric equations used for estimating aboveground biomass stocks of trees and shrubs in different habitats on Mt. Kilimanjaro.

<table>
<thead>
<tr>
<th>Authors</th>
<th>Type</th>
<th>Equation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chave et al. 2005</td>
<td>Dry forest</td>
<td>$\text{AGB} = \exp(-2.187 + 0.916 (\ln(wd \times \text{dbh}^2 \times H)))$</td>
</tr>
<tr>
<td>Chave et al. 2005</td>
<td>Wet forest</td>
<td>$\text{AGB} = \exp(-2.557 + 0.940 (\ln(wd \times \text{dbh}^3 \times H)))$</td>
</tr>
<tr>
<td>Schnitzer et al. 2006</td>
<td>Lianas</td>
<td>$\text{AGB} = \exp(-1.484 + 2.657 (\ln(\text{dbh})))$</td>
</tr>
<tr>
<td>Hairiah et al. 2010</td>
<td>Bananas</td>
<td>$\text{AGB} = 0.03 \times \text{dbh}^{2.11}$</td>
</tr>
<tr>
<td>Hairiah et al. 2010</td>
<td>Coffee</td>
<td>$\text{AGB} = 0.281 \times \text{dbh}^{2.06}$</td>
</tr>
<tr>
<td>Mugasha et al. 2013</td>
<td>Savannas</td>
<td>$\text{AGB} = 0.0763 \times \text{dbh}^{2.2046} \times H^{0.4918}$</td>
</tr>
</tbody>
</table>

Notes: Parameter used for the equations are: wood density (wd, dry weight divided by fresh volume), diameter at breast height (dbh) and total plant height (H).